Water Air Soil Pollut (2010) 211:435–441
DOI 10.1007/s11270-009-0312-5

Point Source Ammonia Emissions are Having a Detrimental Impact On Prairie Vegetation

Carly J. Stevens · David Tilman

Received: 19 October 2009 / Accepted: 7 December 2009 / Published online: 6 January 2010

Abstract Prairie grasslands are very species rich but have declined in their extent considerably due to land-use change and exploitation. Many remaining prairie fragments are situated within an agricultural matrix and can be subjected to high levels of atmospheric ammonia deposition from animal units. Three prairie fragments in Minnesota that were located in close proximity to feedlots were selected, and 500-m transects were studied at an increasing distance from the feedlot. Changes in soil pH, soil nitrate concentration, and soil ammonium concentration with increasing distance from the source were variable between the sites, possibly due to differences in the processing of nitrogen in the soil and the degree of nitrogen limitation. Species richness showed significant negative relationships with ammonia deposition and soil nitrate concentration, whereas aboveground biomass showed a positive relationship with ammonia deposition. Both the richness and biomass of non-graminoid species declined with increasing soil nitrate concentration, whereas graminoid biomass was positively related to ammonia deposition and was negatively associated to richness. *Bromus inermis*, a non-native perennial grass, was the main species that increased at high deposition. The results of this study have important implications for the conservation and restoration of prairie grasslands.

Keywords Ammonia · Soil biogeochemistry · *Bromus inermis* · Nitrogen deposition · Prairie · Species richness

C. J. Stevens (✉)
Department of Life Sciences, The Open University,
Walton Hall,
Milton Keynes MK7 6AA, UK
e-mail: c.j.stevens@open.ac.uk

C. J. Stevens
Lancaster Environment Centre, Lancaster University,
Lancaster LA1 4YQ, UK

D. Tilman
Ecology, Evolution, and Behavior,
University of Minnesota,
100 Ecology Building, 1987 Upper Buford Circle,
St. Paul, MN 55108, USA

1 Introduction

Prairie grasslands support a diverse assemblage of plants, invertebrates, birds, and mammals, but they are under considerable pressure from agriculture and development. The extent of the decline in tallgrass prairie exceeds all other major ecosystems in the USA. Prior to European settlement, prairies covered large portions of the state of Minnesota, but as settlements grew, many were converted to croplands. Now, less than 1% of the original native prairie remains. This usually only survives in areas that are unsuitable for agricultural development, and some of this remaining prairie is subject to exploitation by recreation and overgrazing (Samson and Knopf 1994).

One of the consequences of such extensive habitat loss is that the remaining areas of prairie grasslands are fragmented—scattered through an agricultural matrix. These are frequently small and isolated and can be found in close proximity to intensive agriculture and resultant emissions. Intensive animal units, sometimes housing several hundreds of cows or tens of thousands of poultry, provide a very large source of nitrogen (N) pollution in the form of ammonia volatilized from animal waste. Ammonia concentrations close to animal housing can be large but decline exponentially with increasing distance from the source (Pitcairn et al. 1998). This nitrogen is deposited to the vegetation and soil with the potential to negatively impact nearby ecosystems.

Numerous N addition experiments in different grassland types have shown the potential for N deposition to impact species richness, soil chemistry, and soil microbial processes (e.g., Mountford et al. 1993; Phoenix et al. 2003; Wedin and Tilman 1996). Combined with evidence from regional surveys (Stevens et al. 2004, 2006), there is clear support for a loss of diversity in grasslands that can be linked to atmospheric deposition of N. Long-term studies at Cedar Creek in Minnesota have shown declines in species richness over a period of 24 years in response to low level N applications equivalent to levels of atmospheric N deposition (10 kg N$\,$ha^{-1}year^{-1}; Clark and Tilman 2008). At higher levels of N application (20, 95, and 120 kg N$\,$ha^{-1}year^{-1}), species richness declined after 2 years of N addition with annuals accounting for an increased proportion of the vegetation (Wilson and Tilman 1991a, b). At the same time, the root:shoot ratio of vegetation decreased, productivity increased, and individual species changed in their abundance (Wilson and Tilman 1991a, b). Above- and belowground productivity was elevated throughout the year in all N treatments, and litter showed the same response. As a consequence of the increased productivity, light penetration into the canopy was reduced (Wilson and Tilman 1993). Increases in productivity have been reported, in response, N addition have also been observed in European grasslands (e.g., Wilson et al. 1995), together with increases in species typical of more fertile conditions (e.g., Kirkham and Kent 1997). This is not always the case, and when N is not the limiting nutrient or when the effects of acidification are dominant, there may not be an increase in productivity (e.g., Horswill et al. 2008).

Changes were also seen in the soil with an increase in the availability of nitrate (Wilson and Tilman 1993). Increases in available N with increase in addition or deposition of N have been widely observed (e.g., Emmett et al. 1995; Buchmann et al. 1996; Magill et al. 1997; Stuanes and Kjonaas 1998), although many studies have reported increases in ammonium, but not nitrate, possible due to rapid plant uptake combined with the mobility and lability of any excess nitrate (Matson et al. 2002).

After 6 years of N addition to prairie grassland, the decline in species richness with increasing N addition was still apparent with colonization decreased and loss of species increased with increasing N. Short perennial forbs were particularly sensitive to high inputs including *Solidago nemoralis* and *Hieracium* sp. (Wilson and Tilman 2002). Furthermore, in a larger 12 year study of the impact of N on Minnesota grasslands, Wedin and Tilman (1996) showed a loss of species with increasing N deposition with greatest losses occurring up to 50 kg N$\,$ha^{-1}year^{-1}. In these plots, C_4 grasses were seen to decline, and C_3 grasses increased.

This study will investigate the impact of atmospheric ammonia deposition from animal units on prairie vegetation and soils at three sites in Minnesota using 500-m transects starting close to the ammonia source. The following hypotheses will be tested: (1) Species composition will change with increasing distance from the ammonia source, with greater species richness further away from the ammonia source. (2) Biomass will be higher close to the ammonia source. (3) Soil pH, nitrate, and ammonia concentrations in the soil will change along the gradient of ammonia deposition; pH is expected to be lower as a result of acidification from N deposition, and nitrate and ammonium concentration is higher close to the source.

2 Methods

Three study sites in Minnesota were selected where prairie fragments, consisting of a large proportion of native species and thus of high conservation value, were found in close proximity to animal feedlots.

The first site was an area of prairie grassland between a railway line and a highway near Fairfax, Minnesota. It was on the opposite side of a road, approximately 250 m from a poultry unit housing 10,000 chickens and turkeys. The farm was established in 1990. The prairie extended a kilometer beyond the feedlot, along the road. Dominant species in the prairie grassland included *Rosa arkansana* and *Bromus inermis*. The second site was located in northwest Minnesota, near Brooten. The prairie covered approximately 80 ha. Adjacent to the site, on the other side of a small river, approximately 800 m away, was a feedlot with 150 dairy and beef cattle. This farm is thought to be over 100 years old. Dominant species at this site included *Andropogon geradii* and *Solidago gigantea*. The third site selected was located near Arlington to the south-west of Minneapolis and was also adjacent to a railway. It was approximately 500 m from a large beef unit established in 1997 with 820 cattle and approximately 400 m from an older smallholding with 50 cattle, which is over 100 years old. The prairie grassland was limited in its extent to a length of 400 m. Dominant species included *A. geradii* and *Petalostemon purpureum*. Soils at all three sites were classified as black soils.

At each site, three replicate transects were surveyed. Each transect began at a point where prairie vegetation was close to the animal unit. Samples were collected at 50-m intervals up to 400 m from the start of the transect and a final sample at 500 m. The distance between the animal unit and the start point varied. Vegetation was surveyed in 1×1 m quadrats, all higher plants were identified to a species level and percentage cover recorded. Soils were collected from each quadrat at a depth of 0–10 cm below the litter layer. Biomass samples were collected from strips 10-cm wide and 3-m long. Strips were placed parallel to the transect originating in the center of each quadrat. Samples were sorted into functional groups in the field, and all vegetation was thoroughly dried, and the mass was determined.

Soils samples were air dried and ground to 2 mm. Soil pH was determined using a pH meter with a soil solution of 5 g soil and 25 ml deionized water. Nitrate and ammonium concentrations were determined using a Bran-Luebbe AA3 auto analyzer following extraction by shaking with 1 M KCl.

Levels of deposition of ammonia at each of the sampling points was determined using the atmospheric dispersion model AERMOD (US EPA 2007) with deposition to grassland using a grid size of 25×15 m over the whole sampling areas, and sampling points were modeled individually. Long-term meteorological data from the nearest monitoring stations to each site was used to parameterize the model. Emissions were estimated according to the number of animals currently housed in the unit. Emission factors were based on Battye et al. (1994). Within the model, the cattle feedlots were treated as area sources, and the poultry feedlot was treated as two point sources due to ventilation fans located at one end of the two barns comprising the feedlot. Ventilation rates were assumed to meet state recommended requirements (Janni and Jacobson 2003). In order to compare sites, deposition estimated from the feedlots was added to background deposition taken from the National Atmospheric Deposition (2008).

Data were analyzed using simple regression analysis and regression trees in R (R Development Core Team 2007).

3 Results

3.1 Deposition

Modeled ammonia deposition from the feedlots varied between the three sites depending on the source size and distance between source and sink. Deposition ranged from 0.3 to 2.4, 0.01 to 0.04, and 0.08 to 0.27 kg $\mathrm{N\,ha^{-1}\,year^{-1}}$ at sites one to three, respectively. At sites one and two, deposition showed an exponential relationship with distance from the source. At site three, this relationship was complicated by the presence of two animal units (Fig. 1). Background deposition of ammonia at the three sites was the same (5.14 kg $\mathrm{N\,ha^{-1}\,year^{-1}}$).

3.1.1 Soil Chemistry

Examining all three sites together did not show any significant trends between ammonia deposition and soil chemistry. When the sites were examined individually, site one showed a significant negative relationship between ammonia deposition and pH (r^2=0.57, p<0.05) and a nonsignificant trend towards increasing soil nitrate concentration with increasing ammonium deposition. There was no relationship

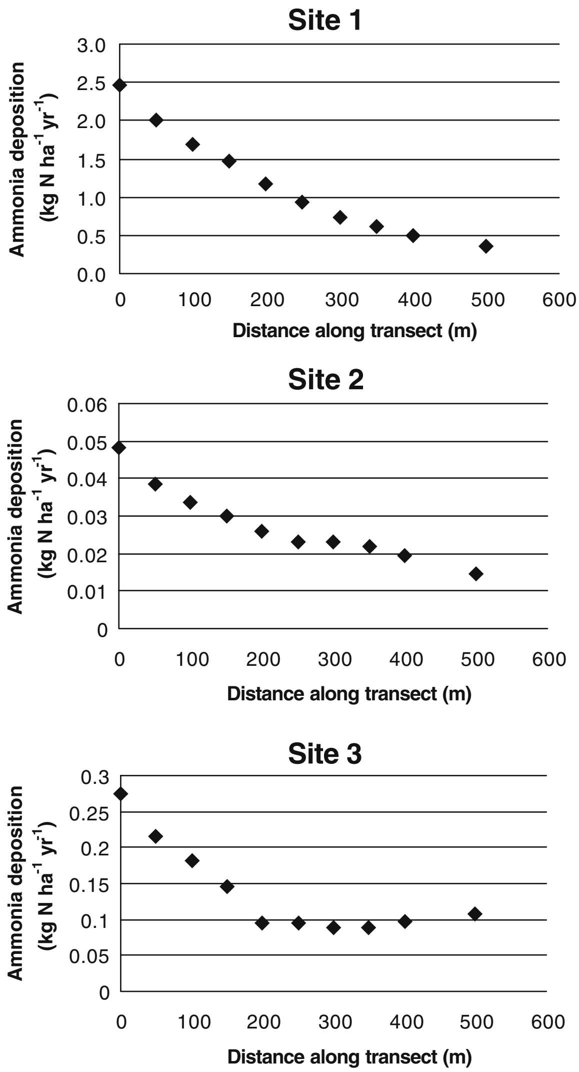

Fig. 1 Modeled ammonia deposition with distance along transect for sites one, two, and three. Size of source, distance of the transect from the source, and position of the source relative to the prevailing wind direction all influence the amount of ammonia deposition

between ammonia deposition and soil ammonia concentration at site one. Site two showed very similar results with a significant negative relationship between deposition and soil pH (r^2=0.77, p<0.001), a nonsignificant trend towards increasing soil nitrate concentration with increasing ammonia deposition, and no relationship with soil ammonium concentration. Site three showed a slightly different response with a nonsignificant trend towards a reduction in soil pH with increasing ammonia deposition and no relationship with soil nitrate concentration. There was, however, a significant relationship between soil ammonium concentration and ammonia deposition (r^2=0.39, p=0.05).

3.1.2 Plant Species Composition

Looking at all three sites together, species richness was weakly negatively correlated with ammonia deposition (r^2=0.22, p<0.01). At individual sites, there was no consistent relationship between ammonia deposition and species richness. Looking at all three sites, there was also a significant relationship between soil nitrate concentration and species richness (r^2=0.29, p<0.01). The best fit for this decline in richness is not a linear relationship as it shows greater losses at low nitrate levels (Fig. 2).

Productivity (total aboveground biomass) showed a positive relationship with ammonia deposition across the three sites (r^2=0.50, p<0.001; Fig. 3). If the outlier at the highest level of deposition is removed, the relationship is greatly improved (r^2= 0.76, p<0.001). Productivity and richness were negatively correlated (r^2=0.35, p<0.001).

Of the soil and deposition variables examined (ammonia deposition, soil nitrate concentration, soil ammonium concentration, and soil pH), graminoid productivity was most closely related to ammonia deposition (r^2=0.35, p<0.001) showing a positive relationship. Graminoid richness was also most closely correlated with ammonia deposition but with a negative relationship (r^2=0.36, p<0.001) indicating an increased dominance of a few graminoid species. The biomass of nongraminoid species (dominated by forbs) was most closely related to soil nitrate

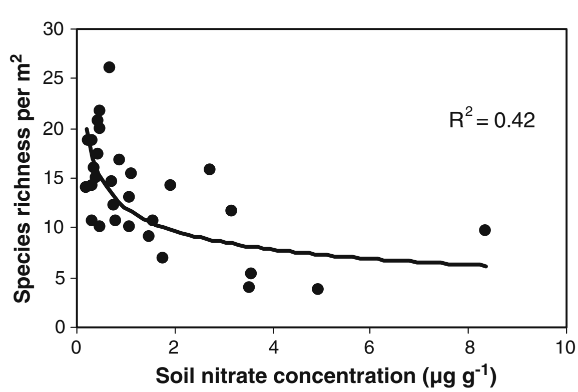

Fig. 2 Mean species richness per 1-m^2 quadrat against mean soil nitrate concentration

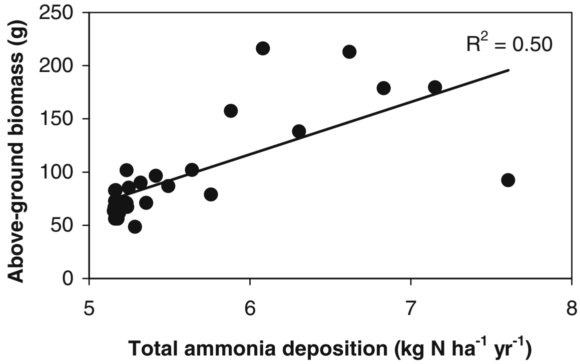

Fig. 3 Mean aboveground biomass against total ammonia deposition (modeled feedlot deposition plus background deposition)

concentration (r^2=0.17, p<0.05) as was the richness of nongraminoid species (r^2=0.31, p<0.01). Both showed a negative relationship with nitrate concentration. Litter biomass showed no consistent trend with increasing N deposition at the three sites.

The regression tree for species richness (Fig. 4) showed the primary importance of ammonia deposition for species richness. It also showed that at higher deposition levels, soil nitrate concentration becomes an important driver of richness indicating an additive effect of nitrate concentration not correlated with ammonia deposition.

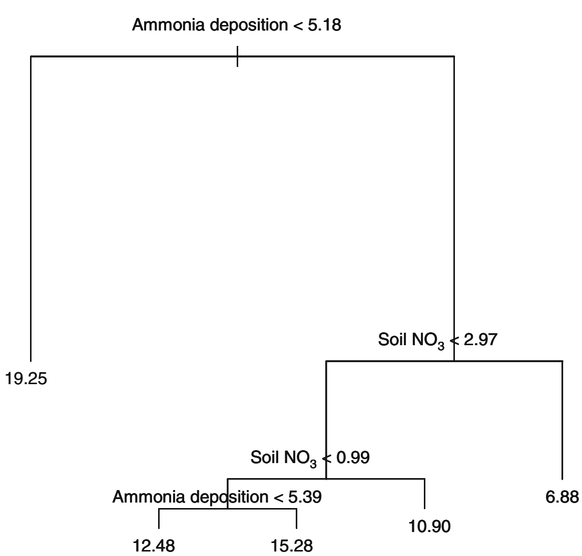

Fig. 4 Regression tree for species richness. Explanatory variables are given together with threshold values separating the response. Values at the end of branches are mean species richness with that category

There were no significant effects of N deposition on the proportion of native species or the proportion of C_4 grasses compared to C_3 grasses.

4 Discussion

4.1 Soil Chemistry

The lack of a consistent response of the nitrate and ammonium concentrations in the soils at the three sites suggests that N cycling and uptake differ at the three sites. This is most likely to be due to different moisture regimes and different degrees of nutrient limitation at the sites. A number of investigations have found that prairie grasslands in this part of the USA are commonly N limited (e.g., Wedin and Tilman 1996). If N is the limiting resource as it is added to the soil, it will be utilized quickly by plants and microbes and will not result in elevated soil concentrations with increasing deposition. The relationship found between ammonia deposition and total aboveground productivity supports this.

Sites one and two show nonsignificant trends towards increasing soil nitrate concentration with increasing ammonia deposition but no relationship between ammonia deposition and soil ammonium concentration. This suggests that the ammonia not taken up by the vegetation is nitrified, converting it to nitrate. This process produces protons acidifying the soil contributing to the significant relationship between soil pH and ammonia deposition. Site three shows a trend for increasing soil ammonium with increasing deposition but no trend with nitrate, suggesting that nitrification is not occurring to the same extent. If this site is more strongly N limited than the other two sites, plant N uptake and/or soil conditions may be limiting nitrification. Less nitrification would also result in less soil acidification.

Soil chemical differences between the sites may also be related to amount of time that it has received elevated deposition. N accumulates in the soil over time (Fowler et al. 2004), and long-term deposition may give a different response than deposition over a short period of time (Skiba et al. 1998). All of the farms in this study had been established for many years, but information on animal numbers and exact times were not available.

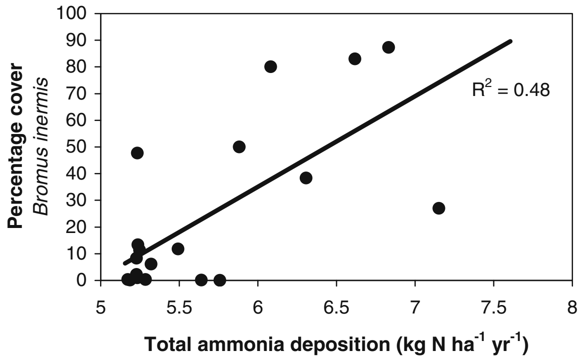

Fig. 5 Mean percentage cover of *Bromus inermis* against total ammonia deposition (modeled feedlot deposition plus background deposition)

4.2 Species Composition

The general trend found across all three sites for reduced species with increasing levels of N deposition and soil nitrate concentration is something that has frequently been observed in prairie grasslands (e.g., Wilson and Tilman 2002; Wedin and Tilman 1996). The increase in productivity has also been observed in similar grasslands leading to reduced light penetration of the canopy (Wilson and Tilman 1993). The increase in available nutrients leads to vigorous growth of those species able to take advantage of the increased availability of N. Species not able to take advantage of these conditions, typically those adjusted to low soil N status, are reduced in their abundance as a result of competition.

In this investigation, the diversity of both grasses and forbs was negatively related to deposition or soil nitrate, but productivity of grasses increased, indicating an increased dominance of a small number of grass species. Examining the percentage cover of individual grass species in relation to ammonia deposition showed that most grass species, including the native grass species, declined in their cover; however, the invasive non-native grass *B. inermis* increased in cover with increasing N deposition (r^2=0.29, p<0.001; Fig. 5). If the outlier at the highest level of deposition is removed, the relationship is again considerably improved (r^2=0.68, p<0.001). *B. inermis* is a fast-growing species that can form a dense litter layer. Another non-native perennial pasture grass, *Agropyron repens*, showed similar increases at elevated N deposition in east central Minnesota (Wedin and Tilman 1996).

The increase in graminoid cover and a loss of nongraminoid species at higher levels of N deposition is a phenomenon that has been observed in a number of habitats worldwide. Stevens et al. (2006) report declines in forb richness and cover in calcifuge grasslands along a national deposition gradient in the UK, while similar trend have been reported in woodland, heathland, and bog communities (e.g., Pitcairn et al. 1998; Heil and Bobbink 1993; Hogg et al. 1995).

This study suggests that a close proximity between prairie grasslands and large animal units is linked to degradation in the quality of prairie grassland fragments. Through a loss of species richness and an increase in non-native species, especially the non-native grasses *B. inermis* (this study) and *A. repens* (Wedin and Tilman 1996), the conservation value of a prairie fragment is reduced. Little of the original prairie of North America remains. For instance, only about 2% of the native prairie of Minnesota is still intact. Our results suggest that protection of these fragments may require practices that prevent locally elevated deposition of ammonium, such as, occurring from nearby feedlots, manure spreading or use of ammonia fertilizers. Large, year-round ammonia sources, such as feedlots, may require sufficient distance between the source and prairie fragments so that deposition levels return to close to background levels. For the three sources we studied, distances of 0.5 to 1.0 km would seem sufficient, but each case would merit explicit modeling. In total, our results suggest that even small elevations in the soil N concentration can lead to changes in productivity and species richness in prairie-grassland communities that should be included in land use planning and conservation.

Acknowledgements A fellowship for CS to visit the University of Minnesota was jointly funded by The British Grassland Society Stapledon Memorial Fellowship and the British Council Researcher Exchange Program. The authors would like to thank J. Sullivan, C. Clark, T. Mielke, G. Brand, A. Ball, R. Hill, C. McFadden, Minnesota Pollution Control, and Minnesota Department of Natural Resources for assistance with this project. Thanks to E. Dorland for comments on an earlier version of this manuscript.

References

AERMOD US EPA (2007) URL: http://www.epa.gov/scram001/dispersion_prefrec.htm. August 2009.

Battye, R., Battye, W., Overcash, C. and Fudge, S. (1994). Development and selection of ammonia emission factors. EPA 68-D3-0034. Research Triangle Park: US EPA Atmospheric Research and Exposure Assessment Laboratory.

Buchmann, N., Gebauer, G., & Schulze, E. (1996). Partitioning of ^{15}N-labeled ammonium and nitrate among soil, litter, below- and above-ground biomass of trees and understory in a 15-year-old *Picea abies* plantation. *Biogeochemistry, 33*, 1–23.

Clark, C. M., & Tilman, D. (2008). Loss of plant species after chronic low-level nitrogen deposition to prairie grasslands. *Nature, 451*, 712–715.

R Development Core Team. (2007). R: a language and environment for statistical computing. Vienna, Austria: R Foundation for Statistical Computing. ISBN 3-900051-07-0, URL: http://www.R-project.org. August 2009.

Emmett, B. A., Brittain, S. A., Hughes, S., Gorres, J., Kennedy, V. H., Norris, D., et al. (1995). Nitrogen additions ($NaNO_3$ and NH_4NO_3) at aber forest, wales: I. Response of throughfall and soil water chemistry. *Forest Ecology and Management, 71*, 45–59.

Fowler, D., O'Donoghue, M., Muller, J. B. A., Smith, R. I., Dragosits, U., Skibal, U., et al. (2004). A chronology of nitrogen deposition in the UK between 1900 and 2000. *Water, air and Soil Pollution Focus, 4*, 9–23.

Heil, G. W., & Bobbink, R. (1993). Impact of atmospheric nitrogen deposition on dry heathlands. A stochastic model simulating competition between *Calluna vulgaris* and two grass species. In R. Aerts & G. W. Heil (Eds.), *Heathlands: patterns and processes in a changing environment*. London: Kluwer Academic Publishers.

Hogg, P., Squires, P., & Fitter, A. H. (1995). Acidification, nitrogen deposition and raid vegetation change in a small valley mire in Yorkshire. *Biological Conservation, 71*, 143–153.

Horswill, P., O'Sullivan, O., Phoenix, G. K., Lee, J. A., & Leake, J. R. (2008). Base cation depletion, eutrophication and acidification of species-rich grasslands in response to long-term simulated nitrogen deposition. *Environmental Pollution, 155*, 336–349.

Janni, K. A., Jacobson, L.D. (2003). Mechanical ventilation for poultry barns. Minnesota: University of Minnesota, Minnesota Extension Service. URL: http://www.bae.umn.edu/extens/poultry_factsheets/mechvent03.pdf. August 2009.

Kirkham, F. W., & Kent, M. (1997). Soil seed bank composition in relation to the above-ground vegetation in fertilized and unfertilized hay meadows on a Somerset peat moor. *Journal of Applied Ecology, 34*, 889–902.

Magill, A., Aber, J. D., Hendricks, J. J., Bowden, R. D., Melillo, J. M., & Steudler, P. A. (1997). Biogeochemical response of forest ecosystems to simulated chronic nitrogen deposition. *Ecological Applications, 7*, 402–415.

Matson, P., Lohse, K. A., & Hall, S. J. (2002). The gloabalization of nitrogen deposition: consequences for terrestrial ecosystems. *Ambio, 31*, 113–119.

Mountford, J. O., Lakhani, K. H., & Kirkham, F. W. (1993). Experimental assessment of the effects of nitrogen addition under hay-cutting and aftermath grazing on the vegetation of meadows on a Somerset peat moor. *Journal of Applied Ecology, 30*, 321–332.

National Atmospheric Deposition Program. (2008). National Atmospheric Deposition Program. URL: http://nadp.sws.uiuc.edu/. August 2009.

Phoenix, G. K., Booth, R. E., Leake, J. R., Read, D. J., Grime, P., & Lee, J. A. (2003). Effects of enhanced nitrogen deposition and phosphorus limitation on nitrogen budgets of semi-natural grasslands. *Global Change Biology, 9*, 1309–1321.

Pitcairn, C. E. R., Leith, I. D., Sheppard, L. J., Sutton, M. A., Fowler, D., Munro, R. C., et al. (1998). The relationship between nitrogen deposition, species composition and foliar nitrogen concentrations in woodland flora in the vicinity of livestock farms. *Environmental Pollution, 102*, 41–48.

Samson, F. B., & Knopf, F. L. (1994). Prairie conservation in North America. *BioScience, 44*, 418–421.

Skiba, U., Sheppard, L. J., Pitcairn, C. E. R., Leith, I. D., Crossley, A., van Dijk, H. F. G., et al. (1998). Soil nitrous oxide and nitric oxide emissions as indicators of elevated atmospheric n deposition rates in seminatural ecosystems. *Environmental Pollution, 102*, 457–461.

Stevens, C. J., Dise, N. B., Gowing, D. J., & Mountford, J. O. (2006). Loss of forb diversity in relation to nitrogen deposition in the UK: regional trends and potential controls. *Global Change Biology, 12*, 1823–1833.

Stevens, C. J., Dise, N. B., Mountford, J. O., & Gowing, D. J. (2004). Impact of nitrogen deposition on the species richness of grasslands. *Science, 303*, 1876–1879.

Stuanes, A. O., & Kjonaas, O. J. (1998). Soil solution chemistry during four years of NH_4NO_3 addition to a forested catchment at Gardsjon, Sweden. *Forest Ecology and Management, 101*, 215–226.

Wedin, D., & Tilman, D. (1996). Influence of nitrogen loading and species composition on the carbon balance of grasslands. *Science, 274*, 1720–1723.

Wilson, S. D., & Tilman, D. (1991a). Component of plant competition along an experimental gradient of nitrogen availability. *Ecology, 72*, 1050–1065.

Wilson, S. D., & Tilman, D. (1991b). Interactive effects of fertilization and disturbance on community structure and resource availability in an old-field plant community. *Oecologia, 88*, 61–71.

Wilson, S. D., & Tilman, D. (1993). Plant competition and resource availability in response to disturbance and fertilization. *Ecology, 74*, 599–611.

Wilson, S. D., & Tilman, D. (2002). Quadratic variation in old-field species along gradients of disturbance and nitrogen. *Ecology, 83*, 492–504.

Wilson, E. J., Wells, T. C. E., & Sparks, T. H. (1995). Are calcareous grasslands in the UK under threat from nitrogen deposition? An experimental determination of a critical load. *Journal of Ecology, 83*, 823–832.